GLOBAL CHANGE BIOLOGY
BIOENERGY

GCB Bioenergy (2017) **9**, 725–742, doi: 10.1111/gcbb.12394

GHG emissions and other environmental impacts of indirect land use change mitigation

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Abstract

The implementation of measures to increase productivity and resource efficiency in food and bioenergy chains as well as to more sustainably manage land use can significantly increase the biofuel production potential while limiting the risk of causing indirect land use change (ILUC). However, the application of these measures may influence the greenhouse gas (GHG) balance and other environmental impacts of agricultural and biofuel production. This study applies a novel, integrated approach to assess the environmental impacts of agricultural and biofuel production for three ILUC mitigation scenarios, representing a low, medium and high miscanthus-based ethanol production potential, and for three agricultural intensification pathways in terms of sustainability in Lublin province in 2020. Generally, the ILUC mitigation scenarios attain lower net annual emissions compared to a baseline scenario that excludes ILUC mitigation and bioethanol production. However, the reduction potential significantly depends on the intensification pathway considered. For example, in the moderate ILUC mitigation scenario, the net annual GHG emissions in the case study are 2.3 $MtCO_2$-eq yr^{-1} (1.8 tCO_2-eq ha^{-1} yr^{-1}) for conventional intensification and -0.8 $MtCO_2$-eq yr^{-1} (-0.6 tCO_2-eq ha^{-1} yr^{-1}) for sustainable intensification, compared to 3.0 $MtCO_2$-eq yr^{-1} (2.3 tCO_2-eq ha^{-1} yr^{-1}) in the baseline scenario. In addition, the intensification pathway is found to be more influential for the GHG balance than the ILUC mitigation scenario, indicating the importance of how agricultural intensification is implemented in practice. Furthermore, when the net emissions are included in the assessment of GHG emissions from bioenergy, the ILUC mitigation scenarios often abate GHG emissions compared to gasoline. But sustainable intensification is required to attain GHG abatement potentials of 90% or higher. A qualitative assessment of the impacts on biodiversity, water quantity and quality, soil quality and air quality also emphasizes the importance of sustainable intensification.

Keywords: agricultural intensification, agriculture, bioenergy, environmental impact assessment, GHG emissions, indirect land use change mitigation and prevention

Received 15 March 2016; accepted 27 May 2016

Introduction

Expanding biofuel production can lead to direct and indirect land use change (DLUC and ILUC). Direct LUC is the change from a previous land use to biofuel feedstock production. Indirect LUC is a change in land use elsewhere because the direct LUC results in either (i) displaced production of agricultural food, feed and fibers to continue to meet the demand, or (ii) more land being taken into agricultural production because of increased food prices (Searchinger *et al.*, 2008; Plevin *et al.*, 2010; Wicke *et al.*, 2012). When ILUC causes conversion of high carbon stock lands such as forests or grasslands, this can lead to greenhouse gas (GHG) emissions which reduce or even cancel out the GHG mitigation potential of biofuels (Searchinger *et al.*, 2008; Plevin *et al.*, 2015; Valin *et al.*, 2015). Although modeling results are characterized by large variation and uncertainty, nearly all studies show significant (I)LUC-related GHG emissions (Wicke *et al.*, 2012; Plevin *et al.*, 2015; Valin *et al.*, 2015). Therefore, mitigation or prevention of ILUC and its impacts are essential for sustainable biofuel production. Through the implementation of different ILUC mitigation measures like agricultural intensification and land zoning, a large amount of additional biofuels can be produced with a low risk of causing ILUC (Wicke *et al.*, 2015; Gerssen-Gondelach *et al.*, 2016). However, besides the reduction in ILUC and associated emissions, the implementation of the ILUC mitigation strategies may also have other environmental impacts (e.g., on GHG emissions from agriculture or on biodiversity). These impacts are not yet well understood.

Agricultural intensification is suggested to play a key role in mitigating ILUC (Erb *et al.*, 2009; van Vuuren *et al.*, 2009; Dornburg *et al.*, 2010; Slade *et al.*, 2011;

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Gerssen-Gondelach *et al.*, 2016). The reason is that agricultural intensification reduces the area of land required for food and feed production, which potentially results in surplus agricultural land that can be used for biomass production. The GHG and environmental impacts of agricultural intensification, however, will significantly depend on the intensification pathway. For example, Valin *et al.* (2013) assess the effects of crop yield and livestock feed efficiency scenarios on the GHG balance of agricultural production in developing countries. They find that when above-baseline gains in crop yield are attained by intensive fertilizer application, the global GHG savings compared to the baseline are about 450 $MtCO_2$-eq yr^{-1} in 2050. In the case of sustainable intensification, that is, through practices that improve crop yields without additional synthetic fertilizer, the emission savings are one-third higher (Valin *et al.*, 2013). In addition, Valin *et al.* (2013) find that, on a global level, improvements in livestock production have a larger effect on GHG mitigation than the intensification of crop production. Yet, they do not investigate the effect of energy crop production expansion on the total GHG balance of the agricultural and bioenergy sector (Valin *et al.*, 2013). This effect is examined by de Wit *et al.* (2014), Melillo *et al.* (2009) and van der Hilst *et al.* (2014). de Wit *et al.* (2014) and Melillo *et al.* (2009) assess the net GHG impacts of bioenergy expansion while mitigating ILUC through agricultural intensification on a European and global scale, respectively. But they include only one intensification pathway for crop production, and they exclude intensification in the livestock sector. Also, their GHG balances are not very detailed as these only account for nitrogen emissions, net soil organic carbon (SOC) fluxes and abated fossil emissions (Melillo *et al.*, 2009; de Wit *et al.*, 2014). van der Hilst *et al.* (2014) perform a more detailed regional study of the net GHG balance in Ukraine. Valin *et al.* (2013) show that such a regional study is important because the GHG impacts of agricultural intensification depend on region-specific factors such as, for example, the degree of intensification possible based on the current yield gap. van der Hilst *et al.* (2014) include two intensification pathways for crops. The second, sustainable intensification pathway includes a few GHG mitigation measures like reduced tillage, but both pathways assume balanced fertilization of crops. Therefore, the impacts of different agricultural intensification pathways, such as the change in GHG emissions due to different nutrient use efficiencies, are not properly evaluated. In other regional case studies that perform a detailed assessment of the GHG balance of bioenergy production, the impacts of agricultural intensification are not, or only partly, taken into account (van Dam *et al.*, 2009; Smeets *et al.*, 2009) or assumed to be negligible because intensification only takes place through improved management (van der Hilst *et al.*, 2012). But in addition to GHG emissions, these case studies do assess other environmental impacts of bioenergy production (van Dam *et al.*, 2009; Smeets *et al.*, 2009; van der Hilst *et al.*, 2012). Agricultural intensification can also have other environmental impacts that can be both positive and negative, and these impacts should be investigated as well. Therefore, the aim of this article was to assess for a specific region and biofuel case (i) the net GHG balance of agricultural and biofuel production under different ILUC mitigation scenarios and (ii) the influence of different intensification pathways on the GHG balance and other environmental impacts. This study is conducted for the case of miscanthus-based ethanol production in Lublin province, Poland.

Materials and methods

Inputs from case study ILUC mitigation

This study builds on a previous study by Gerssen-Gondelach *et al.* (2016) who assessed the low-ILUC-risk production potential of miscanthus-based ethanol in the province of Lublin in 2020. They analyzed five ILUC mitigation measures that reduce the extent of ILUC or control the type of land use change and calculated how these strategies could contribute to the availability of surplus agricultural land on which miscanthus can be produced without causing undesired LUC. As a reference, a baseline scenario was defined for agricultural land use in 2020. This baseline accounted for the agricultural production required to fulfill the projected demand for food, feed and fibers and the mandated production of first generation biofuels in the EU (Laborde, 2011). To assess the ILUC mitigation potential, three scenarios (low, medium and high) were defined in which four ILUC mitigation measures were applied to different degrees. Three of these measures are related to land management: above-baseline yield development, use of underutilized land and land zoning. The fourth measure is improved food chain efficiency (i.e., reduced losses during storage and transportation). A fifth measure, increased integration of the food and biofuel chains, was not applicable to this case study. Table 1 summarizes the assumptions underlying each scenario. In addition, Table 2 presents the resulting land conversions, for example from grassland to miscanthus, in the different scenarios for 2020 compared to 2010. The resulting miscanthus-based ethanol production potential in the three ILUC mitigation scenarios ranged from 16 to 23 PJ yr^{-1} for an average miscanthus yield of 13 t dm ha^{-1} yr^{-1} (Gerssen-Gondelach *et al.*, 2016).

GHG emissions

To assess the impact of ILUC mitigation on the total GHG balance of agricultural and bioenergy production, the annual emissions from these sectors are calculated for 2010 and for

Table 1 Summary of data inputs for 2010 and scenario assumptions for 2020

			ILUC mitigation		
	2010	2020 Baseline	Low	Medium	High
Crops					
Wheat yield (t ha^{-1})*	3.7	4.1	4.5	5.7	7.5
Cattle					
Dairy (milk) productivity (kg product $animal^{-1}$ yr^{-1})	4600	5165	5165	7070	6490
Beef (beef) productivity (kg product $animal^{-1}$ yr^{-1})	200	225	225	315	290
Total cattle density (1000 animals ha^{-1})†	1.5	1.5	1.8	1.8	4.0
Farm characteristics					
Avg. # tractors per farm	0.7	0.8	1.0	1.1	2.1
Avg. farm size (ha)	7	10	15	29	56
Food chain efficiency					
Wheat losses during storage and transport (%)*	5.0	5.0	3.8	2.5	0.8
Biomass production on underutilized land					
Area of underutilized land available for biomass production (1000 ha)	n.a.	n.a.	92	173	375
Land zoning					
Area excluded from biomass cultivation (1000 ha)	n.a.	n.a.	203	236	269

For more information about the assumptions underlying the scenarios, see Gerssen-Gondelach *et al.* (2016); n.a., not applicable.
*As example, scenario assumptions are only presented for wheat.
†The cattle density is expressed as the number of animals per ha of meadows and pastures.

each scenario in 2020. This study does not perform a complete life cycle inventory, but includes the key GHG sources for which the emissions change due to the implementation of ILUC mitigation measures[1] and bioenergy production (Fig. 1). The calculations of the GHG emissions are further explained in the following subsections.

As above-baseline yield developments (i.e., intensification) in crop and livestock production were found to play an important role in ILUC mitigation, it is also assessed how three different intensification pathways in term of sustainability influence the GHG balance of each ILUC mitigation scenario. These pathways are defined as:

- *Conventional intensification (CI):* Yield increases are attained by applying more fertilizers, pesticides and mechanization. The nutrient, pesticide and energy use efficiency are not improved (Valin *et al.*, 2013). Also, conventional agricultural practices such as full tillage are applied.
- *Intermediate sustainable intensification (II):* Yield increases are attained while the nutrient, pesticide and energy use efficiency are enhanced to some extent through improved agricultural practices like reduced tillage.
- *Sustainable intensification (SI):* Yield increases are principally attained without increased input, but through adopting practices which optimize the resource use efficiency and strengthen the productive capacity of the soil (Garnett *et al.*, 2013; Hochman *et al.*, 2013; Valin *et al.*, 2013). These practices include good fertilizer management, integrated pest protection, reduced or no tillage and the prevention of monocultures. Good fertilizer management aims to optimize crop yields while minimizing nutrient losses, for example, using the right nutrient source and applying the fertilizer at the right rate, time and place (Reetz *et al.*, 2015). This could also include the use of improved fertilizer types such as nitrification inhibitors and slow-release fertilizers (Smith *et al.*, 2008, 2014). Integrated pest protection or management integrates all available pest control strategies, including biological, physical and other nonchemical methods, with the aim to minimize the use of chemical pesticides (Möckel, 2015).

Based on the total GHG balances, it is assessed whether the ILUC mitigation scenarios for 2020 abate emissions compared to 2010, the baseline scenario for 2020, and gasoline. Finally, a sensitivity analysis is performed to assess the influence of different assumptions on the results.

Agricultural crops: The projected growth in crop yields increases from the baseline scenario to the high ILUC mitigation scenario. The higher the yield growth, the more advances in the production system are required to attain the projected yields. For example, Fig. 2 presents crop yields of maize, wheat and rapeseed in 2010 and all scenarios in 2020 and shows the

[1]Changes in GHG emissions resulting from increased chain efficiencies in food storage and transport are only partly included. First, as a result of higher efficiencies, food production can be reduced to fulfill the same demand. The changes in GHG emissions due to reduced production are taken into account. Second, efficiency improvements also lead to reduced fuel consumption during storage and transport. However, this is not included because the food losses during storage and transport are generally low (2.5% or less of the total food weight; FAO, 2014; Gerssen-Gondelach et al., 2016) and reductions in these losses will thus have a limited effect on fuel use. As the share of emissions from storage and transport in the total GHG balance of the food chain is considered to be small as well (Moll and Watson, 2009), the reductions in GHG emissions are expected to be negligible.

Table 2 Current land use in agriculture (2010) and projected land conversions in 2020 in the baseline and indirect land use change (ILUC) mitigation scenarios (Gerssen-Gondelach *et al.*, 2016)

				ILUC mitigation		
	1000 ha	2010	2020 Baseline*	Low	Medium	High
Cropland	Remaining cropland	983	755–949	875	740	445
	Conversion to miscanthus	0	0	108	171	189
	Conversion to abandoned land	0	34–229	0	72	350
Grassland†	Remaining grassland	240	189–233	158	114	56
	Conversion to miscanthus	0	0	39	65	80
	Conversion to abandoned land	0	8–51	0	18	51
Underutilized land	Remaining underutilized	50–95	50–95	4	75	95
	Conversion to miscanthus	0	0	46	0	0
	TOTAL‡	1274–1319	1274–1319	1274	1298	1319

*Based on the projected agricultural production and yield developments in the baseline scenario, a large area of cropland and grassland (280 thousand ha) was calculated to be abandoned. However, this is a large reduction in a short time frame. Therefore, in the low and medium ILUC mitigation scenarios, the baseline reduction in agricultural land use is assumed to be lower.

†Meadows and pastures.

‡The total agricultural area in Lublin was 1745 thousand ha in 2010. The agricultural area included here accounts for the most important crops in terms of land use and for meadows and pastures.

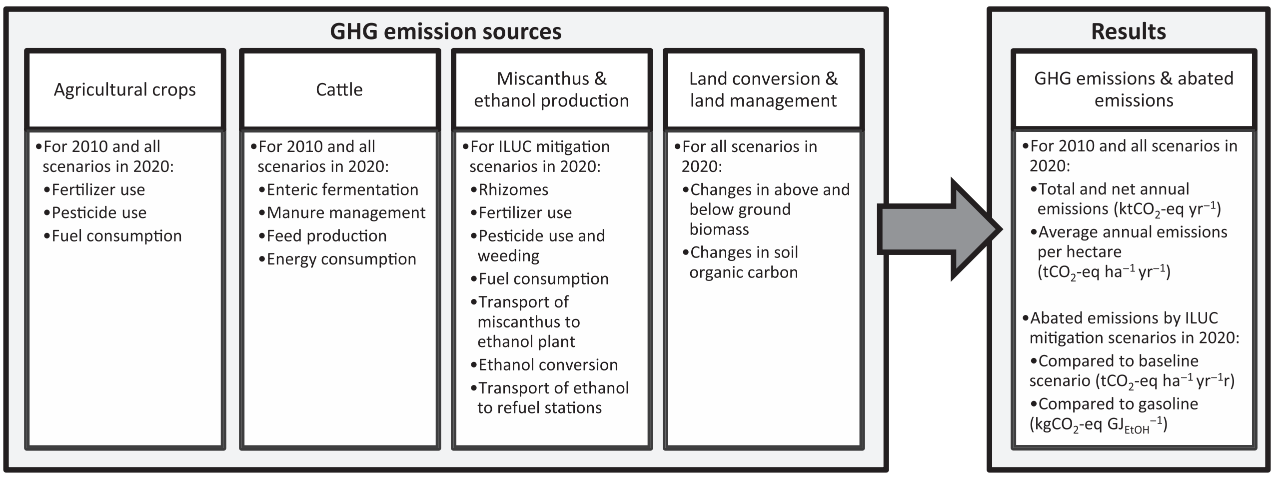

Fig. 1 Overview of GHG emission sources included in the GHG balances and results.

related management level, based on agro-climatically attainable yields for different management systems as described in the Global Agro-Ecological Zones (GAEZ) methodology (FAO, IIASA, 2015). In 2010, the case study region is characterized by an intermediate management level. This means that the production is partly subsistence based and partly market oriented, is medium labor intensive, applies both manual labor, animal traction and some machinery and uses moderate levels of fertilizer and pesticides (FAO, IIASA, 2015). In the medium and/or high ILUC mitigation scenarios, an advanced management level is required which is characterized by commercial production, full mechanization, low labor intensity and optimal use of fertilizers and pesticides (FAO, IIASA, 2015). Below, it is described how the levels of fertilizer, pesticide and fuel consumption are defined in each scenario and how the related GHG emissions are calculated. GHG emissions from seeds are excluded because the amount of seeds is assumed to remain unchanged, while yields increase in the different scenarios.

Fertilizers: This study includes the fertilizer nutrients nitrogen (N), phosphate (P) and potash (K). The emissions related to fertilizer include emissions from fertilizer production, direct N_2O emissions from the soil and indirect N_2O emissions from volatilization, leaching and runoff. For the ILUC mitigation scenarios, the emissions are calculated for the three intensification pathways. For these pathways, different assumptions are made with regard to nutrient use efficiency (NUE) and emission factors for N_2O emissions from N inputs, see below. The measure of nutrient use efficiency applied in this study is the partial factor productivity (PFP), expressed in units of crop yield per unit of nutrient applied (Fixen *et al.*, 2015).

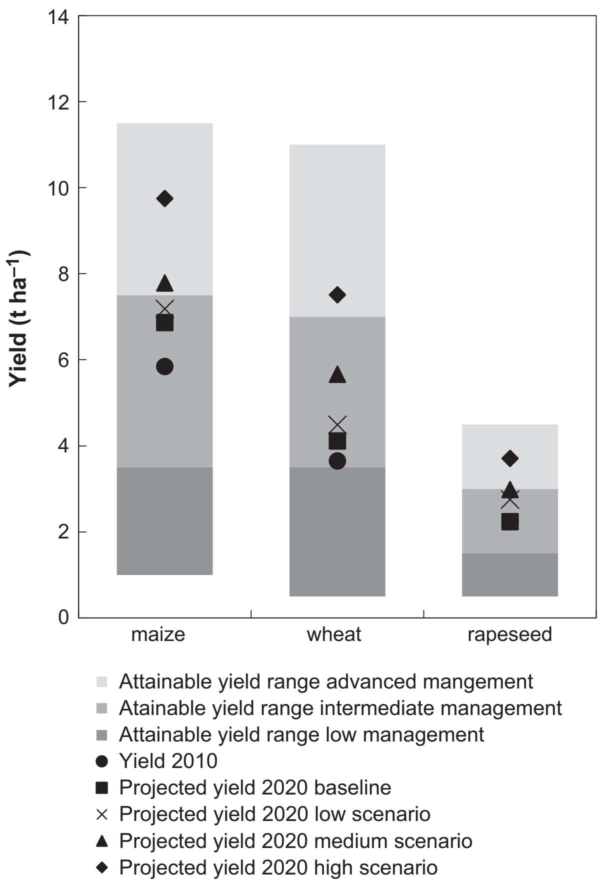

Fig. 2 Maize, wheat and rapeseed yield levels in 2010 and 2020 scenarios and attainable yield ranges for low, intermediate and advanced management levels, based on GAEZ data (FAO, IIASA, 2015). For rapeseed, the yields in 2010 and the baseline scenario for 2020 are equal (data points overlap).

To determine the emissions related to fertilizer application, first the amount of fertilizer consumed is calculated. For three crops, that is, wheat, maize and rapeseed, the fertilizer consumption level is based on the correlation between yield, fertilizer use (in kg ha^{-1}) and NUE, which is derived from historical data points for Lublin, Poland, Germany, the EU and Ukraine (FAO, 2003, 2006, 2007; Rosas, 2012; Heffer, 2013). For example, Fig. 3 presents historical nitrogen use levels in maize cultivation. From the data points, three isolines for constant nutrient use efficiency are derived. These three levels of NUE are designated as low, medium and high. Low NUE reflects suboptimal management conditions and potential overfertilization (Fixen *et al.*, 2015). High NUE is attained through optimized management, but there might also be a risk that the nutrient supply is limiting the crop yield (Fixen *et al.*, 2015).

In the year 2000, the fertilizer levels applied and the crop yields attained in Lublin and Poland were lower compared to Germany and the EU, but the resulting NUE was higher (see data point for Lublin). However, in the following decade, Poland accessed the EU, fertilizer use per hectare increased and the NUE decreased, while the efficiencies in Germany and the EU improved over time (Fig. 3). For Lublin, no data on fertilizer use by crop are available for years after 2000. Therefore, the NUE in Lublin in 2010 is based on the trend in Poland and assumed to be low. Also, in the baseline scenario, this efficiency is assumed to remain low. In the ILUC mitigation scenarios, low NUE is assumed in the case of conventional intensification, medium NUE in the case of intermediate sustainable intensification and high NUE in the case of sustainable intensification.

Based on the calculated fertilizer application per hectare for wheat, maize and rapeseed and the total cultivation area of these crops in Lublin in 2010 (CSO, 2015), the total fertilizer consumption for these crops is calculated. This consumption is subtracted from the total fertilizer use in Lublin in 2010. The remaining fertilizer consumption is divided by the cultivation area of other agricultural crops, resulting in the average fertilizer use per hectare for these crops in 2010. Then, the fertilizer use for these other crops in the baseline scenario is based on the historical (2002–2014) average annual increase in fertilizer use per hectare in Lublin (in % per year relative to 2010). In the ILUC mitigation scenarios, the increase in fertilizer use for the other crops is assumed to develop in the same way as the increases for wheat, maize and rapeseed relative to the baseline scenario.

To calculate the emissions from fertilizer production, emission factors (gCO_2-eq $\mathrm{kg}_{\mathrm{nutrient}}^{-1}$) from the JEC E3-database for biofuel GHG calculations are applied as presented in the BioGrace GHG calculation tool (BioGrace Project, 2015), see Table 3. Direct and indirect N_2O emissions are calculated by applying the IPCC Tier 1 methodology (De Klein *et al.*, 2014) and assuming different emission factors for the three intensification pathways, based on the uncertainty range for default values (Table 3).

Pesticides: Comparable to fertilizers, pesticides use per hectare and crop yields appear to be positively correlated (Schreinemachers & Tipraqsa, 2012). However, for pesticides, not enough historical data are available to derive this correlation per crop as was done for fertilizers. Therefore, based on the limited data available, assumptions are made about low, average and high pesticide use efficiencies. This approach is applied to wheat, rapeseed, apples and the aggregated groups cereals (including oats, triticale, rye, barley, mixed cereals) and other crops (including potatoes, sugar beet and maize). The pesticides levels in 2010 are based on most recent (2002–2008) data for Poland (Surawska & Kołodziejczyk, 2006; Berent-Kowalska & Stobiecki, 2009). Because the management levels in Lublin are lower than the Polish average, it is assumed that the figures for Poland in the period 2002–2008 are appropriate for Lublin in 2010 and represent a medium pesticide use efficiency. For the emission factor of pesticides, see Table 3.

Machinery fuel use: Mechanization is an important driver for increasing yields and thus results in higher fuel use. Lorencowicz & Uziak (2009) assessed the fuel consumption at family farms in Lublin and found that the fuel consumption is linearly correlated to the farm size (in ha). They derived relationships for different groups of farms: farms with only one tractor,

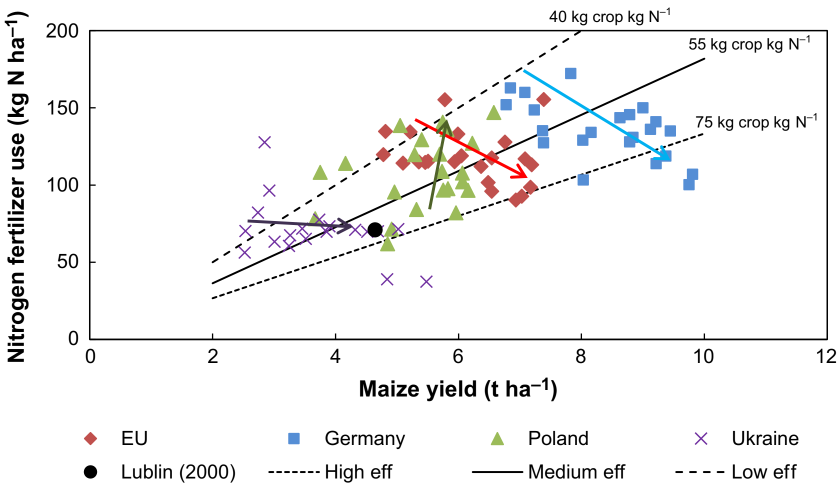

Fig. 3 Historical maize yields and fertilizer application levels in Lublin, Poland, Germany, EU and Ukraine (1990–2010). Isolines indicate the correlation between yield and fertilizer level for constant nutrient use efficiencies (low: 40 kg crop kg N^{-1}, medium: 55 kg crop kg N^{-1}, high: 75 kg crop kg N^{-1}). Arrows indicate the direction in yield development, fertilizer level and nutrient use efficiency over time in Poland, Germany, EU and Ukraine.

Table 3 Key values for the calculation of GHG emissions from agricultural production

Process	Aspect	Value	Unit	References
General	GWP CH_4	25	g CO_2-eq g CH_4^{-1}	
	GWP N_2O	298	g CO_2-eq g N_2O^{-1}	
Cultivation	Fertilizer production, N	5881	g CO_2-eq kg N^{-1}	JEC E3-database BioGrace Project (2015)
	Fertilizer production, P	1011	g CO_2-eq kg P^{-1}	Idem
	Fertilizer production, K	576	g CO_2-eq kg K^{-1}	Idem
	Direct N_2O emission factor*	0.015/0.01/0.006	kg N_2O–N kg $N_{applied}^{-1}$	IPCC De Klein *et al.* (2014)
	Volatilization fraction*	0.15/0.1/0.06	kg NH_3–N + NO_x–N kg $N_{applied}^{-1}$	Idem
	Volatilization emission factor*	0.02/0.01/0.006	kg N_2O–N kg NH_3–N + NO_x–N^{-1}	Idem
	Leaching and runoff fraction*	0.5/0.3/0.2	kg N kg $N_{applied}^{-1}$	Idem
	Leaching and runoff emission factor*	0.015/0.0075/0.0050	kg N_2O–N kg $N_{leached}^{-1}$	Idem
	Pesticide production	10 971	g CO_2-eq kg a.i.$^{-1}$	JEC E3-database BioGrace Project (2015)
	Diesel	38.3	MJ l^{-1} (HHV)	Hydrogen Analysis Resource Center (2015)
		87.6†	g CO_2-eq MJ_{diesel}^{-1}	JEC E3-database BioGrace Project (2015)

GWP, global warming potential; a.i., active ingredient.
*Values for, respectively, conventional/intermediate sustainable/sustainable intensification.
†Emission factor for diesel includes indirect emissions.

farms with more than one tractor, and all farms. In 2010, the average amount of tractors per farm in Lublin was less than 1 (Table 1). This is projected to remain true in the baseline scenario for 2020. For these cases, the formula derived for farms with only one tractor is applied to calculate the fuel use. In the ILUC mitigation scenarios, the average number of tractors per farm is projected to be 1 or higher than 1, but there may be a mix of farms with one or more tractors. Therefore, for these scenarios, the formula for all farms is applied and assumed to be true for intermediate sustainable intensification. For conventional and sustainable intensification, a higher and lower fuel consumption levels are estimated. The emission factor for diesel is presented in Table 3. The production of machinery is not included in the emission balance.

Cattle: The total emissions from dairy and beef production include emissions from enteric fermentation (CH_4), manure management (CH_4 and N_2O), feed production (N_2O and CO_2) and energy consumption (CO_2). For 2010 and each scenario, as well as for the three intensification pathways in each ILUC mitigation scenario, the emissions for each component (in kg CO_2-eq kg_{milk}^{-1} or kg CO_2-eq kg_{beef}^{-1}) are estimated based on a review of case studies by Gerssen-Gondelach *et al.* (2015a) and FAO data on average emissions from enteric fermentation and manure management in European countries and regions (FAO, 2015). Then, the total annual emissions are calculated by multiplying the emissions per kg product by the total annual milk or beef production. Cattle production in 2010 was primarily characterized by a pasture-based production system. In this system, some of the feed comes from grazing on pastures and a larger part from fodder (CSO, 2015). However, grasslands are often extensively managed and their productivity is lower compared to some Western European regions because of the lower annual precipitation (Scientific Committee on Animal Health and Animal Welfare, 2001; Sanderson *et al.*, 2013). Therefore, forage maize constitutes the largest part of the feed mix (Scientific Committee on Animal Health and Animal Welfare, 2001). With increasing animal productivity and density in the scenarios for 2020 (see Table 1), the feed quality is initially improved by more intensive management of meadows and pastures, followed by including more feed crops and concentrates in the feed mix and reducing the time at pasture. Thus, the importance of grazing decreases, similar to dairy production in Germany (Reijs *et al.*, 2013). The three intensification pathways in each ILUC mitigation scenario illustrate how GHG emissions are influenced by different levels of energy use efficiency and nutrient use efficiency in pasture management and feed production, and by different manure management practices. As meadows, pastures and forage maize were not included in the GHG balance of agricultural crops, no double counting occurs when including feed production in the GHG balance of cattle in 2010 and the baseline scenario. In the ILUC mitigation scenarios, the share of crops in the feed mix increases, but is assumed that the effect of potential double counting on the total GHG balance will be small because the share of emissions from feed production is limited to 5–12% of the total emissions from cattle. In addition, the emissions from feed production also include manure applied to feed crops and pasture or directly deposited on pastures, which is not included in the GHG balance of agricultural crops.

Miscanthus and ethanol production: In this study, a lifetime of 20 years for miscanthus is assumed. This lifetime includes 1 year of establishment followed by 19 years of spring harvesting (van der Hilst *et al.*, 2012). The average yield is 13 t dm ha^{-1} (Gerssen-Gondelach *et al.*, 2016).

Rhizomes: Rhizomes are planted in the year of establishment (Smeets *et al.*, 2009). Regarding the emission factor for rhizomes, a wide range is found in the literature (Smeets *et al.*, 2009; van der Hilst *et al.*, 2012). This range is used to define emissions factors for the three intensification pathways, see Table 4.

Fertilizer application: Miscanthus requires less fertilizer than annual crops. It is characterized by efficient nitrogen use and a large part of the nutrients is translocated to the roots during the senescence of the leaves during winter (Lewandowski *et al.*, 2000). In the literature, there is no consensus on the response of miscanthus yield to fertilization rates (Smeets *et al.*, 2009; Haines *et al.*, 2015). Therefore, the amount of nutrient (N, P and K) removed with harvested biomass is used as a proxy for the required fertilizer application level (Smeets *et al.*, 2009; Cadoux *et al.*, 2012). Recommended application rates range between 3.0–4.9 kg N t dm^{-1}, 0.47–0.6 kg P t dm^{-1} and 6.5–7.0 kg K t dm^{-1} (Smeets *et al.*, 2009; Cadoux *et al.*, 2012). These ranges are used to define the application levels for the different intensification pathways. For nitrogen and potash, the resulting fertilizer consumption per hectare is in line with field studies in Lublin (Borzęcka-Walker, 2010; Matyka & Kus, 2011; Borzęcka-Walker *et al.*, 2012; Borkowska & Molas, 2013). But for phosphate, the calculated application rate is significantly lower than reported in the literature (Borzęcka-Walker, 2010; Matyka & Kus, 2011; Borkowska & Molas, 2013). Therefore, the impact of the phosphate level is assessed in the sensitivity analysis. Miguez *et al.* (2008) state that miscanthus cultivation may result in lower nitrate leaching compared to annual crops, but the fraction of N lost through leaching and runoff is uncertain (Cadoux *et al.*, 2012). Therefore, for each intensification pathway, this study assumes lower emission factors compared to those applied for agricultural crops but still within the uncertainty range from the IPCC Tier 1 methodology (De Klein *et al.*, 2014), Table 4.

Pesticides: Pesticides, primarily herbicides, are only applied during the establishment phase of the plantation and after removal of the plantation at end of the production cycle (Smeets *et al.*, 2009). In addition, weeding is applied occasionally in early years (Borkowska & Molas, 2013) and is assumed unnecessary in later years due to increasing plant density. Because of the limited pesticide use, the application level is kept constant for the three intensification pathways.

Fuel use: During miscanthus cultivation, machinery is used for soil preparation, planting of rhizomes, fertilizer application, weeding, harvesting (including baling) and removing the plantation at the end of the plantation's lifetime (Smeets *et al.*, 2009). Additional fuel is consumed for storage of the bales, transport of bales to the bioethanol plant and transport of ethanol to fuel stations (Smeets *et al.*, 2009). For all operations except final transport, the required machinery, work capacity (h ha^{-1}) and fuel use are derived from Smeets *et al.* (2009) and EcoInvent datasets (Nemecek & Kägi, 2007), Table 4. For final ethanol transport to the fuel stations, three options are considered depending on the transportation distance. First, in case ethanol is used locally, transport can be performed by truck (Hamelinck *et al.*, 2005). This option is included in the ILUC mitigation scenarios. Second, in the case ethanol is exported, train transport or shipping can be chosen. The effects of these

Table 4 Key values for the calculation of GHG emissions from the miscanthus and bioethanol value chain

Process	Aspect	Value	Unit	References
Cultivation	Miscanthus rhizomes*	280/200/110	kg CO_2-eq ha^{-1}	Smeets *et al.* (2009) and van der Hilst *et al.* (2012)
	Fertilizer use, N*	5.0/4.0/3.0	kg N t dm^{-1}	Idem
	Fertilizer use, P*	0.6/0.55/0.5	kg P t dm^{-1}	Idem
	Fertilizer use, K*	7.0/6.5/6.0	kg K t dm^{-1}	Idem
	Direct N_2O emission factor*	0.01/0.006/0.003	kg N_2O–N kg $N_{applied}^{-1}$	IPCC De Klein *et al.* (2014)
	Volatilization fraction*	0.1/0.06/0.03	kg NH_3–N + NO_x–N kg $N_{applied}^{-1}$	Idem
	Volatilization emission factor*	0.01/0.006/0.001	kg N_2O–N kg NH_3–N + NO_x–N^{-1}	Idem
	Leaching and runoff fraction*	0.3/0.2/0.1	kg N kg $N_{applied}^{-1}$	Idem
	Leaching and runoff emission factor*	0.0075/0.005/0.0025	kg N_2O–N kg $N_{leached}^{-1}$	Idem
	Diesel consumption, small tractor (60 kW)*	12/7.5/5	l h^{-1}	Smeets *et al.* (2009) and Nemecek & Kägi (2007)
	Diesel consumption, medium tractor (75 kW)*	20/15/7.5	l h^{-1}	Idem
	Diesel consumption, large tractor (100 kW)*	22/20/15	l h^{-1}	Idem
Storage of bales	Biomass loss	3	% dm	Monti *et al.* (2009), Shinners *et al.* (2010) and Smeets *et al.* (2009)
	Fuel use	0.9	l t dm^{-1}	Smeets *et al.* (2009)
Truck transport	Truck transport, max load	27	ton	Idem
	Average distance	50	km^{-1}	Assumption
	Fuel empty	0.2	l km^{-1}	Smeets *et al.* (2009)
	Fuel full	0.4	l km^{-1}	Idem
Biomass conversion to bioethanol	Primary energy use	0.1	MJ_P MJ_{EtOH}^{-1}	Hoefnagels *et al.* (2010)
		34	g CO_2-eq MJ_P^{-1}	Idem
	Electricity cogeneration	0.1	MJ_{Elec} MJ_{EtOH}^{-1}	Hamelinck & Faaij (2006), Aden *et al.* (2002) and Tao & Aden (2009)
Truck transport to fuel station	Maximum load†	25	ton	Hamelinck *et al.* (2005)
	Diesel consumption	18.1	MJ km^{-1}	Idem
	Average distance	100	km	Idem
Train transport to fuel station	Maximum load†	1000	ton	Idem
	Electricity use	163	kWh km^{-1}	Idem
	Average distance	800	km	Idem
Ship transport to fuel station	Maximum load†	4000	ton	Idem
	Diesel consumption	647	MJ km^{-1}	Idem
	Average distance	1100	km	Idem

(continued)

Table 4 (continued)

Process	Aspect	Value	Unit	References
Energy	Electricity, EU mix (2009)	125.5	g CO_2-eq MJ_{elec}^{-1}	BioGrace Project (2015)
	Electricity, Polish mix (2009)	285.9	g CO_2-eq MJ_{elec}^{-1}	Idem
	Ethanol	23.4	MJ l^{-1} (HHV)	Oak Ridge National Laboratory (2008)
		0.794	ton m^{-3}	BioGrace Project (2015)

*Values for, respectively, conventional/intermediate sustainable/sustainable intensification.
†The maximum load is restricted by mass; thus, the maximum volume is not entirely utilized.

options are assessed in the sensitivity analysis. The estimates on rail and shipping distances are based on the assumption that ethanol is exported to Western European countries (Hamelinck *et al.*, 2005), Table 4.

Biomass conversion to bioethanol: The conversion of miscanthus to ethanol requires chemicals and energy, but also cogenerates electricity, see Table 4. To account for electricity cogeneration in the GHG balance, system expansion is applied. Thus, the GHG balance includes the savings of not generating electricity in a conventional power plant. This credit is based on the Polish electricity mix, which heavily relies on coal-based power generation (Table 4).

Land conversion and land management: Carbon stock changes take place due to land conversion and changes in land management. In this case study, five types of land conversion are considered (Table 2). Carbon stock changes are calculated according to the IPCC Tier 1 approach (Lasco *et al.*, 2007; Verchot *et al.*, 2007) and the guidelines as published in the EU Commission Decision (2010), Eqns (1)–(3).

$$\Delta CS = CS_A - CS_R \tag{1}$$

$$CS_i = SOC + C_{BM} \tag{2}$$

$$C_{BM} = \left(\frac{Y}{HI} - Y\right) \times CF_B \tag{3}$$

where ΔCS = carbon stock change due to land conversion, CS_i = carbon stock associated with land use i (t C ha^{-1}), A = actual land use, R = reference land use, SOC = soil organic carbon (t C ha^{-1}), C_{BM} = carbon stock in aboveground and belowground biomass (t C ha^{-1}), Y = yield (t dm ha^{-1}), HI = harvest index, based on the harvestable yield compared to the total aboveground and belowground biomass, CF_B = carbon fraction of dry matter in biomass (t C ha^{-1}).

Changes in the carbon stock of dead organic matter (DOM) are excluded as DOM stocks are considered zero for nonforest land (European Commission, 2010). For 2010 and the baseline scenario, region-specific data on the SOC stocks of croplands and grasslands are used (Borzecka-Walker, 2014; Lugato *et al.*, 2014). For other land uses, the SOC is a function of the standard SOC (SOC_{ref}) specified for the applicable climate and soil type multiplied by three factors related to the land use (F_{LU}), soil management (F_{MG}) and carbon inputs (F_I), Eqn (4) and Table 5 (Lasco *et al.*, 2007; European Commission, 2010).

$$SOC = SOC_{ref} \times F_{LU} \times F_{MG} \times F_I \tag{4}$$

With regard to the ILUC mitigation scenarios, it is assumed that the SOC of each land use can change due to altered management practices, depending on the intensification pathway. In the case of intermediate sustainable intensification, the SOC values remain equal to 2010 and the baseline. For conventional and sustainable intensification, the SOC values are assumed to, respectively, decrease or increase by 4–5% based on IPCC (Lasco *et al.*, 2007; Verchot *et al.*, 2007) and annual mitigation potentials for these lands as estimated by Smith *et al.* (2008). In absolute terms, the carbon stock change is 0.3 t C ha^{-1} yr^{-1} for croplands and 0.4 t C ha^{-1} yr^{-1} for grasslands.

Net GHG balances and GHG abatement levels: To attain the net annual emissions from agricultural and bioenergy production in Lublin in 2010 and all scenarios in 2020, the annual emissions from all GHG sources as discussed in the previous sections are added up. Based on these net GHG balances, it is assessed whether the ILUC mitigation scenarios abate emissions compared to 2010 and the baseline scenario for 2020. In addition, when the net emissions are fully included in the GHG balance of bioenergy, it is assessed whether the production of bioethanol in the ILUC mitigation scenarios abates emissions compared to gasoline and whether the GHG reduction levels comply with EU GHG savings requirements (European Commission, 2015). The emission factor for gasoline is 90 g CO_2-eq $MJ_{gasoline}^{-1}$ (Fritsche *et al.*, 2009).

Sensitivity analysis: To assess how the results are influenced by different assumptions on, for example, the electricity mix and the ethanol chain efficiency, a sensitivity analysis is performed for the medium ILUC mitigation scenario with intermediate sustainable intensification. Table 6 gives an overview of all variables included and the assumptions made in the sensitivity analysis.

Table 5 Default values for the calculation of carbon stock changes due to land conversion

Process	Aspect	Specification	Value	Unit	References
General	Climate	Cool temperate, moist			
	$SOC_{cropland}$	Average	75	t C ha^{-1}	Borzecka-Walker (2014)
	$SOC_{grassland}$	Average	88	t C ha^{-1}	Borzecka-Walker (2014)
	SOC_{ref}	Clay soils	95	t C ha^{-1}	European Commission (2010)
Arable crops	C_{BM}		0	t C ha^{-1}	Idem
Apples	F_{LU}	Perennial	1	Factor	European Commission (2010) and Lasco *et al.* (2007)
	F_{MG}	Full tillage	1	Factor	Idem
	F_I	High input without manure	1.11	Factor	Idem
	$C_{BM,\ apples}$		43.2	t C ha^{-1}	European Commission (2010)
Grassland	$C_{BM,\ grass}$	Grassland	6.8	t C ha^{-1}	Idem
Abandoned land	F_{LU}	Set-aside	0.82	Factor	Lasco *et al.* (2007)
	F_{MG}	No tillage	1.15	Factor	Idem
	F_I	Low input	0.92	Factor	Idem
Miscanthus	F_{LU}	Grassland*	1	Factor	European Commission (2010)
	F_{MG}	Improved*	1.14	Factor	Idem
	F_I	Medium input*	1	Factor	Idem
	HI		0.4	Factor	Himken *et al.* (1997) and van der Hilst *et al.* (2012)
	CF_B		0.47	t C tdm^{-1}	European Commission (2010)
Equilibrium time	D	Time needed to reach equilibrium soil C stock	20	year	Idem

SOC, soil organic carbon; C_{BM}, carbon stock in aboveground and belowground biomass; F_{LU}, land use factor; F_{MG}, management factor; F_I, input factor; HI, harvest index; CF_B, carbon fraction of dry matter in biomass; D, equilibrium time.
*Perennial herbaceous crops like miscanthus are not included as a separate land use in the guidelines of the IPCC (Lasco *et al.*, 2007) and European Commission (2010). The factors of improved grassland and medium input are assumed to be most appropriate for miscanthus, see van der Hilst *et al.* (2012).

Table 6 Assumptions for sensitivity analysis of the medium ILUC mitigation scenario (intermediate sustainable intensification pathway)

	Variable	Medium scenario: original assumptions	Sensitivity low	Sensitivity high
Miscanthus–ethanol chain	Phosphate level adapted to field studies in Lublin	7 kg P ha^{-1}	–	33 kg P ha^{-1}
	Biomass conversion to bioethanol, electricity credit	Electricity mix Poland (285.9 g CO_2-eq MJ_{elec}^{-1})	Electricity mix EU27 (125.5 g CO_2-eq MJ_{elec}^{-1})	–
	Truck distance	100 km	50 km	150 km
	Transport mode	Truck	Train	Ship
	Chain efficiency (miscanthus and ethanol yield)	13 t dm ha^{-1}, 84.4 GJ_{EtOH} ha^{-1}	10 t dm ha^{-1}, 63.6 GJ_{EtOH} ha^{-1}	17 t dm ha^{-1}, 128.7 GJ_{EtOH} ha^{-1}
	Equilibrium time (LUC)	20 year	10 year	40 year

Assessment of other environmental impacts

The ILUC mitigation scenarios should not only save GHG emissions compared to fossil fuels, but should also have a positive or at least neutral impact on other parts of the environment. This means that the implementation of the ILUC mitigation measures and the cultivation and processing of miscanthus must not be at the expense of biodiversity, ground and surface water quantity and quality, soil quality and air quality (van Dam *et al.*, 2009; Franke *et al.*, 2012). This study qualitatively assesses the risk that this principle cannot be met. Based on literature, several indicators are selected to discuss and evaluate the potential changes in biodiversity, water quantity and quality, soil quality and air quality for the three intensification pathways in 2020. First, with regard to biodiversity, a distinction is made between areas of high nature value (HNV) and other areas. For HNV areas, this study evaluates the extent to which it is possible to continue meeting the habitat functions requirements of species living in these areas (van der Hilst *et al.*, 2012). For other areas, it is assessed how their species abundance might be affected by changes in land use and land management (van Dam *et al.*, 2009; van der Hilst *et al.*, 2012). Second, with regard to water, the water availability is discussed based on the rates of precipitation and evapotranspiration (Smeets *et al.*, 2009; van der Hilst *et al.*, 2012). The water quality is related to the risk of leaching of fertilizers and pesticides (van der Hilst *et al.*, 2012). Third, the soil quality is considered by assessing the risk of reducing the productive capacity of the soil and the risk of erosion (Franke *et al.*, 2012; van der Hilst *et al.*, 2012). Here, soil organic carbon (SOC) is used as a proxy indicator for the productive capacity of the soil (van der Hilst *et al.*, 2012). Soil erosion includes wind and water erosion (Smeets *et al.*, 2009; Franke *et al.*, 2012; van der Hilst *et al.*, 2012). Salinization is excluded, as the risk of this type of erosion is considered negligible in Lublin (Tóth *et al.*, 2008). Finally, indicators for air quality are emissions of non-GHG pollutants causing acidification (SO_2, NH_3 and NO_x) and emissions of fine particles (PM_{10}) (Franke *et al.*, 2012). Also, the risk of contamination of the air due to pesticides is discussed. The potential impact of each intensification pathway on these indicators is evaluated based on literature. Also, the expected impacts are qualified using symbols, ranging from – for a high risk of negative effects to ++ for no risk and high positive effects. It should be noted that this qualification is not always straightforward as it reflects the interpretation of the authors.

Results

GHG emissions

The annual GHG balances for 2010 and 2020 are presented in Fig. 4. In the baseline scenario for 2020, the net annual emissions are slightly lower compared to 2010. Emissions from machinery and cattle are reduced in the baseline scenario, but emissions from fertilizers increase. Carbon stock changes due to land conversion and changes in land management are negligible. In 2010 and the baseline scenario, the largest GHG emission sources are fertilizer use on cropland and enteric fermentation from cattle production. The ILUC mitigation scenarios generally reduce emissions compared to the baseline, see also Fig. 5a. The only exception is the low-ILUC mitigation scenario following the conventional intensification pathway. This can especially be explained by reductions in the SOC stocks of all land uses due to the application of conventional management practices. These land use related soil carbon emissions are larger than the total carbon sequestration related to the conversion of agricultural lands to miscanthus, which results in positive net carbon emissions. In the medium and high ILUC mitigation scenarios following the conventional intensification pathway, the negative emissions related to LUC are higher and counteract the land use related reduction in carbon stocks, resulting in negative net carbon emissions.

It is found that the differences between net annual GHG emissions and GHG abatement levels are often larger for intensification pathways in the same ILUC mitigation scenario than for ILUC mitigation scenarios following the same intensification pathway. The main reason for this is that the intensification pathways have considerable influence on the emissions related to fertilizer consumption and soil carbon stock changes in each land use type. First, the emissions from fertilizer consumption significantly depend on the nutrient use efficiency and emission factors applied. Therefore, the emission reduction attained through improved nutrient use efficiencies and emission factors in a more sustainable intensification pathway is considerably higher compared to the GHG mitigation attained through increased yields and a reduction in the cropland area in a higher ILUC mitigation scenario. Second, the

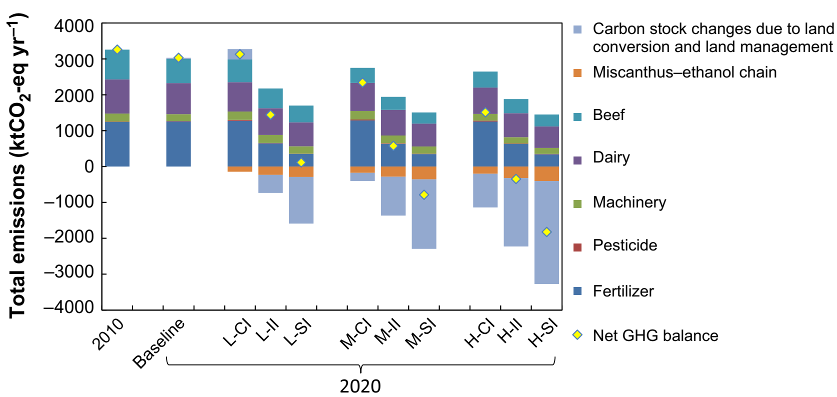

Fig. 4 Total and net annual GHG emissions for 2010 and the baseline and indirect land use change (ILUC) mitigation scenarios in 2020. Emissions from the miscanthus–ethanol value chain are based on truck transport of ethanol to local fuel stations. The equilibrium time for soil carbon stock changes is 20 years. ILUC prevention scenarios: L, low; M, medium; H, high. Intensification pathways: CI, conventional intensification; II, intermediate sustainable intensification; SI, sustainable intensification.

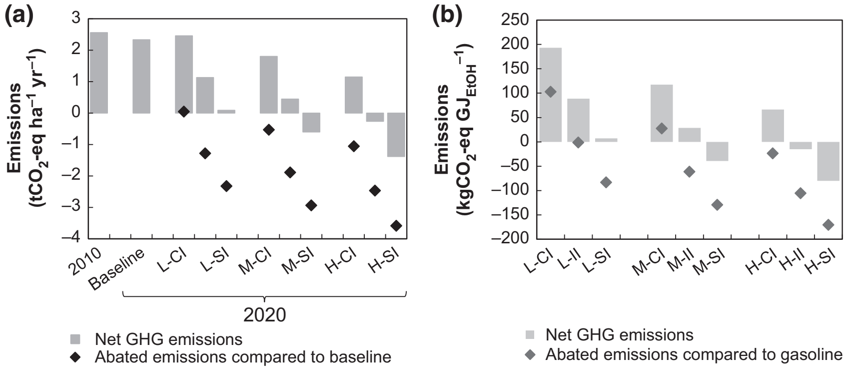

Fig. 5 Net GHG emissions and abated emissions compared to (a) baseline scenario (tCO_2-eq ha^{-1} yr^{-1}) and to (b) gasoline ($kgCO_2$-eq GJ_{ETOH}^{-1}). Positive abatement values mean higher emissions and no emission abatement compared to the reference, negative abatement values mean lower emissions and emission abatement compared to the reference. Indirect land use change (ILUC) prevention scenarios: L, low; M, medium; H, high. Intensification pathways: CI, conventional intensification; II, intermediate sustainable intensification; SI, sustainable intensification.

difference in SOC stocks between the intensification pathways due to different management practices is generally larger than the difference in LUC-related carbon stock changes between the ILUC mitigation scenarios.

In the miscanthus–ethanol chain, especially fertilizer use but also storage and transport of bales contribute most to the GHG emission balance. However, due to the high emission factor for the Polish electricity mix, the credit for cogenerated electricity as by-product of ethanol production is very large, which results in negative emissions for the bioethanol chain. Also, land conversion results in carbon sequestration in the ILUC mitigation scenarios. Considering an equilibrium time of 20 years, soil carbon sequestration due to land conversion to miscanthus is found to be 1.5–1.8 t C ha^{-1} yr^{-1} for cropland, 1.0–1.1 t C ha^{-1} yr^{-1} for grassland and 1.2–1.4 t C ha^{-1} yr^{-1} for abandoned land and is highest for the sustainable intensification pathways. These figures are in line with results from de Wit *et al.* (2014) and Borzęcka-Walker *et al.* (2011). The total carbon sequestration in soil and biomass ranges from about 22 to 45 t C ha^{-1}. Only for apple orchards, land conversion to miscanthus causes a significant biomass loss, resulting in carbon emissions of 26–37 t C ha^{-1}. On croplands and grasslands that are abandoned and not used for miscanthus cultivation, the total carbon sequestration in biomass and soil is 6–9 and −12 to −13 t C ha^{-1}, respectively.

When considering the net GHG emissions from the ILUC mitigation scenarios per Gigajoule (GJ) of ethanol produced, it is found that in most cases, miscanthus-based ethanol abates emissions compared to gasoline (Fig. 5b). Exceptions are the low and medium ILUC mitigation scenarios following the conventional intensification pathway. For these cases, the net emissions are, respectively, 114% and 31% higher compared to gasoline. In the low scenario with intermediate sustainable intensification and the high scenario with conventional intensification, the GHG emission reduction compared to gasoline is 1% and 26%, respectively, which is not in compliance with the EU GHG savings requirements of 35% reduction now and 60% reduction in 2018 compared to fossil fuels (European Commission, 2015). The five remaining cases comply with these goals, and four of these also fulfill the even higher reduction objective of 80–90% as suggested by Cramer *et al.* (2007). The latter four cases include all ILUC mitigation scenarios following the sustainable intensification pathway and the high scenario following the intermediate sustainable intensification pathway. The medium and high ILUC mitigation scenarios that attain a negative net GHG balance due to carbon sequestration in biomass and soil even realize an emission reduction of more than 100% compared to gasoline.

Sensitivity analysis. Figure 6 presents the results of the sensitivity analysis. A significant effect is found for the electricity credit from ethanol production, the equilibrium time for carbon stock changes and the miscanthus–ethanol chain efficiency. For these three variables, the impact is highest for the emission abatement level compared to gasoline. The phosphate application rate for miscanthus production and the truck distance and transport mode to deliver bioethanol to fuel stations have negligible impact on the GHG emission abatement level compared to the baseline scenario and to gasoline (less than 1% change) and are therefore not included in Fig. 6.

The equilibrium time is found to have the largest impacts on the results. Increasing the equilibrium time from 20 to 40 years has no influence on which cases reduce emissions compared to the baseline, but the difference between the moderate ILUC mitigation scenario

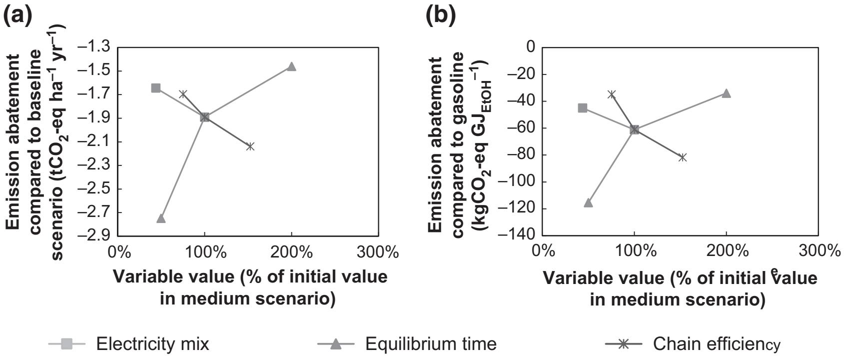

Fig. 6 Sensitivity analysis of the abated emissions in the medium indirect land use change (ILUC) mitigation scenario following the intermediate sustainable intensification pathway: (a) compared to the baseline scenario and (b) compared to gasoline.

following the conventional pathway and the baseline scenario will become very small. Also, the difference in net annual GHG emissions between the three ILUC mitigation scenarios will become smaller. With regard to GHG emission reductions compared to gasoline, fewer cases will attain the GHG savings requirements of 60% or higher. Emission reductions of more than 100% will only be attained by the moderate and high ILUC mitigation scenarios following the sustainable intensification pathway.

In this sensitivity analysis, the electricity credit from ethanol production was changed by applying the EU electricity mix instead of the Polish mix. Of course, the electricity credit also depends on the amount of electricity cogenerated. For example, when electricity cogeneration is halved, the total credit is also reduced to half of the initial credit. The effect on to the net GHG emissions and the emission abatement level compared to the baseline will be equal compared to halving the emission factor for the electricity mix. However, when the reduction in electricity cogeneration is accompanied with a higher conversion efficiency from miscanthus to ethanol, the GHG emission savings can be allocated to a larger amount of ethanol, which decreases the reduction in the emission abatement level compared to gasoline.

Qualitative assessment of other environmental impacts

Key environmental impacts are discussed in the following sections, more information can be found in the Appendix S1.

Biodiversity. In unprotected agricultural areas, intensification is expected to lead to scaling up of farms and potentially also specialization. In the case of conventional intensification, this causes an increase in monocultures and loss, modification and fragmentation of habitats. This, in turn, leads to a decrease in species abundance. Also, inefficient fertilizer and pesticide usage results in increased leaching, soil degradation and water pollution, which are risk factors for species abundance (van der Hilst *et al.*, 2012).

As miscanthus is an extensively managed crop, the conversion of arable lands to miscanthus is often found to have a positive impact on species abundance (Smeets *et al.*, 2009; Dauber *et al.*, 2010). The impacts of the conversion of grasslands to miscanthus are more uncertain and not yet well understood (Dauber *et al.*, 2010; Donnelly *et al.*, 2011), but the risk of biodiversity loss is estimated to be higher. This is especially because many grasslands are seminatural and extensively managed (Sanderson *et al.*, 2013) and the conversion of extensive pastures has a higher risk of biodiversity loss than the conversion of intensively managed pastures (van der Hilst *et al.*, 2012).

As Poland supports a significant share of farmland bird populations in the EU (Sanderson *et al.*, 2013), special attention should be paid to the conservation of these birds. Risk factors for the abundance and sometimes also the species richness of farmland birds are reductions in low-intensity farmland cover (Sanderson *et al.*, 2013) and land conversion of areas with high bird densities to miscanthus cultivation (van der Hilst *et al.*, 2012).

Water. In Poland, annual precipitation is low, but monthly precipitation is highest during summer when the evapotranspiration is also peaking. However, the evapotranspiration exceeds the precipitation and water deficits occur during the summer (Mioduszewski, 2014). In Lublin, the average annual precipitation ranges between approximately 500 and 600 mm depending on the location (Statistical Office in Lublin, 2014). Figures from the literature (Table 7) show that the water requirements of agricultural crops may exceed the annual precipitation, which results in a negative annual

Table 7 Crop evapotranspiration coefficient K_c (dimensionless) and water use efficiency for miscanthus and agricultural crops

	K_c growth period (low/high)	Duration growth period (days)	Water requirement growth period (mm)	WUE (g_{DM} kg^{-1}_{water})	References
Miscanthus (spring harvest)	0.85–1.2	365	n/a	n/a	van der Hilst *et al.* (2012)
Miscanthus (autumn harvest)	0.3–1.6	215	900	4.2	Triana *et al.* (2015)
Wheat	0.25–1.15	100–170	450–650	0.69–0.86	van der Hilst *et al.* (2012) and Dornburg *et al.* (2008)
Maize	0.3–1.2	100–170	500–800	0.70–1.41	Idem
Potato	0.5–1.15	100–175	500–700	1.08–1.89	Idem
Sugar beet	0.35–1.2	160–215	550–750	1.00–1.60	Idem
Pasture	0.70–1.05	200	n/a	n/a	Szejba (2012)
Meadows (extensive)	0.76–1.41	200	n/a	n/a	Idem

K_c is the ratio between the actual nonwater limited water demand and the reference evapotranspiration (ET_0); n/a, not available.

water balance and the occurrence of droughts. Conventional agricultural intensification increases the risk of groundwater deficiencies and droughts (Mioduszewski, 2014). Risk factors include monocultures and irrigation (Smith *et al.*, 2014). Sustainable intensification practices that improve soil moisture, for example, by increasing soil organic carbon through reduced tillage, may help crops to better withstand droughts (Franke *et al.*, 2012; Smith *et al.*, 2014).

Despite its high water use efficiency, the rate of evapotranspiration of miscanthus is found to be higher compared to traditional annual crops and pastures (Smeets *et al.*, 2009; van der Hilst *et al.*, 2012), see Table 7. Large-scale cultivation of miscanthus, especially in the case of monocultures, will thus contribute to the risk of groundwater depletion. In the assessment of the low-ILUC-risk miscanthus potential by Gerssen-Gondelach *et al.* (2016), minimal water requirements (550 mm yr^{-1}) were already taken into account. Thus, areas with a high risk of depleting water bodies and competition with other water uses are excluded from miscanthus cultivation. However, more site-specific data and analysis are required to assess the impacts of miscanthus production on water availability in Lublin. This should take into account variables like the soil texture, rainfall pattern, wind speed, cropping pattern and the location-specific crop evapotranspiration factors for arable crops and miscanthus (Smeets & Faaij, 2010).

Soil. The GHG balances for the ILUC prevention scenarios showed that the chosen intensification pathway has a significant influence on the SOC balance and on the net annual emissions. This is confirmed by the literature. For example, Squire *et al.* (2015) find that increased fertilizer and pesticide use in the UK resulted in reduced SOC and also in lower water-holding capacity of the soil. Thus, sustainable intensification practices are important to prevent SOC losses (Franke *et al.*, 2012). In addition to reduced or no tillage, measures to increase SOC include the use of cover crops and replanting native vegetation on abandoned land (Smith *et al.*, 2008, 2014; Möckel, 2015).

Based on average SOC values used in the GHG emission calculations, the conversion of croplands and grasslands to miscanthus improves the soil carbon stocks. But the variation in SOC values for grasslands is large. Extensively managed grasslands have higher SOC stocks and the risk of carbon loss increases when converting these lands (van der Hilst *et al.*, 2012). The risk of converting high carbon stock pastures increases as more pastures are converted in the higher scenarios. Although grasslands with the highest risk of carbon losses (i.e., boggy and wet areas) were already excluded from miscanthus cultivation in the underlying study determining the surplus land area available for miscanthus (Gerssen-Gondelach *et al.*, 2016), it needs to be carefully assessed which of the remaining grasslands can be converted to miscanthus while achieving SOC sequestration and which grasslands should be maintained to prevent SOC losses.

Air. Important sources of non-GHG pollutants causing acidification are fertilizer use (NH_3), manure management (NH_3) and tractor fuel combustion (NO_x) (Galbraith *et al.*, 2006). Also, pesticides have the potential to contaminate the air and pose a risk on the environment and human health (Sattler *et al.*, 2007). Thus, sustainable intensification, including increased fertilizer, pesticide and fuel use efficiency and adoption of improved manure management technologies, is important to maintain or improve the air quality.

Synthesis. Based on the discussion in the previous sections, Table 8 presents a qualification of the environmental impacts for the three intensification scenarios. It shows that conventional intensification could

Table 8 Potential impacts on current levels of biodiversity, ground and surface water quantity and quality, soil quality and air quality in the case of conventional, intermediate sustainable and sustainable intensification toward 2020

	Risk factor	Conventional intensification	Intermediate sustainable intensification	Sustainable intensification
Biodiversity				
Habitat functions in HNV areas	Land conversion to miscanthus*	−	+/−	+
	Agricultural intensification	+/−	0	+
Species abundance in non-HNV areas	Agricultural intensification	–	+/−	+
	Cropland conversion to miscanthus	+/−	+	++
	Grassland conversion to miscanthus	–	−	+/−
Water				
Water quantity	Agricultural and miscanthus production	–	−	±
Water quality	Agricultural intensification	–	+/−	+
	Miscanthus cultivation	+/−	+	++
Soil				
SOC	Management and conversion of agricultural land	+/−	+	++
Soil erosion	Water erosion	−	+/−	+
	Wind erosion	+/−	+/−	+
Air				
Air quality	Airborne emissions of non-GHG pollutants causing acidification	–	−	+
	Emissions of PM_{10}	−	+/−	+
	Pesticides	–	−	+

Symbols: –, high risk of negative effects; −, considerable risk of negative effects; 0, low risk, no effects expected; +/−, some risk, impacts may be either positive or negative; +, no risk, positive effects expected; ++, no risk, high positive effects expected.
*Only in selected areas where miscanthus cultivation is expected to have a positive impact on biodiversity.

especially pose high risks on biodiversity, water quantity and quality and air quality. In the case of sustainable intensification, impacts are positive for almost all indicators.

Discussion

In this article, we conducted a region- and biofuel-specific assessment of the annual GHG emissions and environmental impacts of agricultural and biofuel production for three ILUC mitigation scenarios (low, medium and high) and for three intensification pathways in terms of sustainability. By investigating the impacts of both agricultural intensification and biofuel production, we approached the subject of ILUC mitigation and biofuel production in a novel, integrated manner. While already a few previous studies have investigated the net GHG balance of biofuel and agricultural production, our approach is more detailed in terms of the GHG emission sources included, the impacts covered, the impacts of different intensification pathways and different ILUC mitigation (or biofuel potential) scenarios. This approach allows giving more comprehensive insights into the GHG and other environmental impacts of biofuel production. Nevertheless, several uncertainties exist and local monitoring of the impacts is recommended.

GHG impacts

The calculations of GHG emissions rely on many uncertain data and assumptions. First, to project fertilizer consumption in 2020, correlations between fertilizer level and crop yield were derived from historical data. Yet, the number of data points found varies significantly between crops and fertilizer types. Therefore, the underpinning of assumptions about ranges in nutrient use efficiencies varies considerably. Also, the actual application levels will depend on local biophysical conditions. To determine fertilizer requirements more exactly, an environmental assessment model could be used that takes into account site-specific agro-ecological circumstances and crop-specific nutrient demand (such as the MITERRA model used in de Wit *et al.*, 2014). Second, with regard to N_2O emission factors, the assumptions are based on IPCC values and might not fully correspond with the local situation. Third, statistical data on pesticide use by crop are very limited and range significantly between countries (Fernandez-Cornejo *et al.*, 2014; CBS, 2015; Fera, 2015). The projections from this article are within these wide ranges, but it is difficult to assess the suitability of the projected pesticide levels specifically for Lublin in 2020. Based on the ranges in pesticide use efficiency specified for different crops, the average annual increase in pesticide consumption is considered to be 0.06 kg a.i. ha^{-1} yr^{-1} in the case of sustainable intensification, 0.14 kg a.i. ha^{-1} yr^{-1} in the case of intermediate sustainable intensification and 0.40 kg a.i. ha^{-1} yr^{-1} in the case of conventional intensification. For comparison, the recent (2005–2012) national average growth trend in the total agricultural sector is 0.09 kg a.i. ha^{-1} yr^{-1} (FAO, 2015). Fourth, in the present study, surplus agricultural land that is not converted to miscanthus cultivation is considered as abandoned land. In the case of cropland, this means that the resulting soil carbon sequestration is lower compared to the sequestration level in case of land conversion to miscanthus. In the case of grasslands with high carbon stocks, the conversion to abandoned land may even result in soil carbon emissions. However, the land that is now assumed to be abandoned could also be converted to other land uses and this would impact the carbon stocks. For example, afforestation or other revegetation could, especially in the long run, significantly increase the carbon stocks in soil and biomass (see, e.g., Schierhorn *et al.*, 2013). In the past 10 years, afforestation in Lublin has been limited to a few hundred hectares per year. But substantial improvements in agriculture, a reduction in agricultural land use or financial support from the government, can potentially increase the afforestation rate.

Other environmental impacts

The qualitative assessment of environmental impacts other than GHG emissions provides a first indication of these effects, but further analysis and quantification are needed. First, the impacts will depend on the local biophysical context. Because of high spatial heterogeneity in biophysical and climate conditions, an improved qualitative assessment could, for example, be carried out by applying our approach to a spatially explicit analysis of environmental impacts. Van der Hilst *et al.* (2012) provide an illustrative example of spatially explicit analysis, which could be further developed to integrate impacts of agricultural intensification. Second, for our assessment, we made a selection of environmental indicators. However, much more indicators exist (see, e.g., McBride *et al.*, 2011). The inclusion of more or other indicators may alter the outcomes of our analysis. For example, considering the evapotranspiration rate, miscanthus is not performing well compared to other crops. But with regard to water exports off-site, miscanthus may be advantageous compared to other crops, particularly when miscanthus is only harvested in spring when its moisture content is low. Third, additional measures to improve environmental impacts, for example, the use of cover crops, may positively affect biodiversity and soil quality. Also, in addition to miscanthus, other biomass crops may be used for energy and environmental services. Cultivation of these crops can alter and potentially further improve the environmental impacts of ILUC mitigation and biofuel production.

Social and economic aspects of ILUC mitigation

Our study focused on the GHG and environmental impacts of ILUC mitigation. Other dimensions of sustainable bioenergy production, that is, the social and economic prerequisites and impacts of ILUC mitigation, are not included and need to be addressed in future research. This should also include an assessment of the trade-offs between environmental and socioeconomic impacts (see, e.g., Smeets & Faaij, 2010). Finally, our previous study on ILUC mitigation potentials (Gerssen-Gondelach *et al.*, 2016) found that agricultural intensification is an important measure to attain high potentials for low-ILUC risk bioethanol production. In addition to that result, our present article shows that the pathway of agricultural intensification has a considerable influence on the GHG and other environmental impacts of ILUC mitigation and biofuel production. Earlier, we already discussed the challenges and barriers for agricultural intensification in general in Lublin, for example, the need for investments (Gerssen-Gondelach *et al.*, 2016). To realize sustainable intensification, supplementary challenges exist and additional measures need to be implemented. In a study on historical yield developments, Gerssen-Gondelach *et al.* (2015b) find that the implementation and enforcement of agri-environmental policies play an important role in sustainable intensification of the agricultural sector. Such policies are aimed at, for example, balanced use of fertilizers, and other inputs or enhanced quality of degraded agricultural lands. In addition, policies should adopt an integrated perspective on all land uses, whether for food, feed, fiber and fuels. Such an integrated approach allows to select and implement measures for each land use which are complementary to each other and optimize the environmental effects of ILUC mitigation and biofuel production.

Acknowledgements

This research was conducted within the research program 'Knowledge Infrastructure for Sustainable Biomass', which is funded by the Dutch Ministries of 'Economic Affairs' and 'Infrastructure and the Environment'.

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Supporting Information

Additional Supporting Information may be found online in the supporting information tab for this article:

Appendix S1. Qualitative assessment of other environmental impacts.